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Effects of forests on particle number concentrations in near-road environments across three geographic regions[☆]

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ABSTRACT

Trees and other vegetation have been advocated as a mitigation measure for urban air pollution mainly due to the fact that they passively filter particles from the air. However, mounting evidence suggests that vegetation may also worsen air quality by slowing the dispersion of pollutants and by producing volatile organic compounds that contribute to formation of ozone and other secondary pollutants. We monitored nanoparticle (>10 nm) counts along distance gradients away from major roads along paired transects across open and forested landscapes in Baltimore (USA), Helsinki (Finland) and Shenyang (China) – i.e. sites in three biomes with different pollution levels – using condensation particle counters. Mean particle number concentrations averaged across all sampling sites were clearly reduced (15%) by the presence of forest cover only in Helsinki. For Baltimore and Shenyang, levels showed no significant difference between the open and forested transects at any of the sampling distances. This suggests that nanoparticle deposition on trees is often counterbalanced by other factors, including differing flow fields and aerosol processes under varying meteorological conditions. Similarly, consistent differences in high frequency data patterns between the transects were detected only in Helsinki. No correlations between nanoparticle concentrations and solar radiation or local wind speed as affecting nanoparticle abundances were found, but they were to some extent associated with canopy closure. These data add to the accumulating evidence according to which trees do not necessarily improve air quality in near-road environments.

1. Introduction

Detrimental effects of atmospheric particle pollution is considered as one of the biggest challenges to human health in urban areas (EEA, 2017; OECD, 2012). Because of their large surface area, planting trees is often advanced as a way to improve urban air quality by filtering particles from the air. However, the initial optimism has turned into skepticism as many modeling and empirical studies have revealed that the relationship between vegetation and air quality is more complex: vegetation may also worsen air quality by reducing ventilation, thereby slowing the dispersion of pollutants by increasing their residence time at street level (Tong et al., 2015; Vos et al., 2013), and by producing volatile organic compounds (VOCs) that contribute to the formation of ozone and other secondary pollutants (Gromke and Ruck, 2012; Pataki et al., 2011; Rantala et al., 2016) as well as new particle formation (e.g. Kulmala et al., 2013). Studies concerning the performance of vegetation in improving air quality have approached the issue in 4 ways: city-scale deposition modeling (e.g. Nowak et al., 2006), local scale empirical studies of ambient concentrations (Hagler et al., 2012) and deposition on vegetation (e.g. Wang et al., 2008), wind tunnel experiments (e.g. Roupsard et al., 2013), and progressive local scale modeling approaches (Karttunen et al., 2020; e.g. Kurppa et al., 2018). These studies report, to some extent, contradictory results due to varying approaches in both scale and methodology (Janhäll, 2015). While evidence of vegetation effects on near-road air quality has matured greatly over the last several years with accumulating empirical data and more

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sophisticated models (e.g. Abhijith et al., 2017; Abhijith and Kumar, 2019; Baldauf, 2017; Barwise and Kumar, 2020; Ghasemian et al., 2017; Tiwari et al., 2019; Viippola et al., 2018; Xing et al., 2019; Yli-Pelkonen et al., 2020), the issue is far from being fully understood and generalizations that apply across the range of conditions encountered in roadside environments are difficult.

Fine particulate pollution ($PM_{2.5}$, particles with aerodynamic diameter less than 2.5 μm) is associated with a wide range of negative health effects (Pope and Dockery, 2006; Silva et al., 2013). Particles less than 100 nm in aerodynamic diameter (commonly referred to as nanoparticles [NP] or ultrafine particles [UFP]) are suspected to have a substantial negative health effects due to their potential to deposit in the respiratory system, cross cell membranes, and be translocated throughout the body (HEI Review Panel, 2013; Maher et al., 2016). NPs from traffic are emitted as primary pollutants from combustion and wear processes of vehicle brakes and also as secondary aerosols formed as emitted gases react to form new particles (Kukutschová et al., 2011; Rönkkö et al., 2017). Regulatory limits to particulate matter (PM) are typically expressed in terms of mass/unit air volume, not count concentrations. However, it is important to recognize that while NPs are so small that their contribution to PM mass is negligible, they dominate particle number concentration (PNC) and have greater surface area per unit volume of atmosphere, which raises their potential to be absorbed and translocated throughout the body (HEI Review Panel, 2013).

Effects of vegetation on NPs emitted from road traffic have not been extensively studied. Al-Dabbous and Kumar (2014) reported significant PNC reductions downwind of a near-road vegetation barrier. Furthermore, whether or not the effect of vegetation is negative or positive has been shown to depend on vegetation structure such as height and thickness (Baldauf, 2017; Deshmukh et al., 2019; Hagler et al., 2012; Janhäll, 2015; Tong et al., 2016). Only a few studies have focused on continuous tree stands and how pollutants are affected by location within a stand. While Viippola et al. (2018, 2016) reported lower coarse particle levels in forest transects than in open, these authors found that samples in forested transects had *higher* NO_2 and PAH levels, and similar $PM_{2.5}$ levels compared to adjacent treeless areas. The vegetation effect have also been shown to depend on the physical properties of each pollutant: gaseous pollutant concentrations either increased or remained unaffected while the largest particles were lower in near-road stands (Setälä et al., 2013; Yli-Pelkonen et al., 2017a). Part of the inconsistencies among results may be due to different climate and vegetation regimes, different pollutant levels of the locations and especially to the varying topographical and vegetation design characteristics.

Here we measured NPs in short campaigns on transects next to major roads across paired open (no tree canopy) and forested landscapes in three urban regions at divergent biomes and pollution levels using condensation particle counters. Our main objective was to find out if there are any generalizable differences in the concentration patterns of NPs between the two landscapes, and commonalities among 3 radically different urban regions. Based on earlier findings regarding gaseous and particle (fine and coarse) pollution in similar environments, we pose these specific hypotheses:

i) Near the road PNC will be higher under tree canopy compared to open area due to restricted dispersion (see e.g. Yli-Pelkonen et al., 2017b),
ii) Further away from the road the pollutant levels will be lower in the forest as the polluted air is not transported as effectively under tree canopy as it is in the open,
iii) Decay curve of NPs will be steeper in the forest due to the accumulation of pollutants at the edge of the forest and weaker transport into the interior due to reduced wind speed.

2. Materials and methods

We measured PNC simultaneously at adjacent open and forest transects situated next to busy roads. Measurements were conducted in three urban regions with dissimilar climates, vegetation and traffic flow: Baltimore metropolitan area (USA), Helsinki metropolitan area (Finland) and city of Shenyang (China) (Table 1). Baltimore with its 2.7 million inhabitants is the economic center of the state of Maryland; Helsinki has ca. 1.5 million inhabitants and is Finland's most significant economic region; Shenyang is the capital of the northeastern province of Liaoning, with a population of ca. 8.1 million people. Measurement campaigns were carried out between May 2015 and May 2017 (see Table S1 in Supplementary Material [SM] for detailed sampling times and dates).

2.1. Instrumentation

We used two portable condensation particle counters (TSI CPC-

Table 1
Sampling distances at each study site measured from the forest edge (forest transects) or from equivalent distances from the road (open transects) and from the roadside, canopy closure estimates (%), traffic volumes and site coordinates. See Figures S1 and S2 in SM for site aerial photographs. Official traffic volumes (motor vehicles day^{-1}) are presented as average annual daily traffic (BAL1–3, HEL1 and HEL3) and weekday average daily traffic for HEL2 (Finnish Transport Agency, Maryland Department of Transportation). On-site traffic volumes are calculations from the sites presented as motor vehicles per minute.

	Sampling distances (m)											
	from the forest edge				from the roadside				Canopy cl.	Traffic		
	1	2	3	4	1	2	3	4	estimate (%)	Official[a]	On-site[b]	Coordinates
Baltimore												
BAL1	Edge	10	25	65	15	25	40	80	86	50,900	61	39°37′23.0″N 76°40′02.0″W
BAL2	Edge	10	25	65	15	25	40	80	83	68,300	85	39°02′30.0″N 76°40′19.9″W
BAL3	Edge	10	25	65	10	20	35	75	85	68,300	77	39°02′28.0″N 76°40′16.0″W
Helsinki												
HEL1	Edge	10	25	65	10	20	35	75	77	62,200	56	60°09′45.1″N 24°51′56.6″E
HEL2	Edge	10	28	73	7	17	35	80	56	32,200	37	60°11′28.4″N 24°55′23.3″E
HEL3	Edge	10	25	65	15	25	40	80	75	24,300	34	60°53′54.1″N 25°36′02.1″E
Shenyang												
SHE1	Edge	10	25	95	5	15	30	100	52	n/a	137	41°46′47.8″N 123°26′18.0″E
SHE2	Edge	20	40	120	10	30	50	130	61	n/a	37	41°46′41.4″N 123°27′56.4″E

[a] Motor vehicles day^{-1}.
[b] Motor vehicles minute^{-1}.

3007, TSI Incorporated, Shoreview, USA) to obtain simultaneous number concentration data along parallel transects at each site (see below). The cutoff range (50% of the particles detected) of the instrument is 10 nm and upper limit 1000 nm with a measurement accuracy of ±20%. We used the sampling frequency of 1 s in order to obtain the most detailed possible view on the PNC dynamics in real time. The highest values close to roads often exceeded the maximum measurable concentration for the instrument (100,000 particles/cm^3), and are considered unreliable. Accordingly, values exceeding 100,000 were reduced to 100,000 when calculating site mean values. This method causes the least possible bias on the data, as excluding these values entirely would skew the habitat comparison. Percentages of events over the upper limit in open and forested transects were 1.3% and 1.7% respectively (Baltimore), 0.2% and 0.1% (Helsinki) and 3.1% and 1.1% (Shenyang).

To verify uniform accuracy between the two instruments, they were operated side by side in the beginning, middle and in the end of each sampling period. This parallel sampling showed that the two instruments give mostly indistinguishable readings (e.g. in Baltimore difference between particle counter median values were 1% ± 7%, AVE ± SD, n = 18). When available, parallel sampling data was used so that each instrument's data were regressed against both instruments' average data values, and the obtained adjustment equations (from linear trendline) were used to eliminate inherent differences among the instruments (see SM Table S2 for correlation coefficients and median differences (%) before and after data correction). The most comprehensive parallel sampling was conducted in Baltimore, while in Shenyang and Helsinki the quality control relied more on repeated flow and zero measurements. Flow measurements show the volume of air pumped through the instruments, which correlates with particle counts, and zero measurement are done with a filter that excludes all particles yielding an approximation of a zero value. Parallel data were unavailable from HEL1 and HEL2, and in HEL3 the instruments were rotated through both transects during sampling, which effectively controlled for possible bias.

2.2. Site information

The study sites, each containing both open and forest transects, were established in the temperate forest biome (Baltimore, humid coastal climate; Shenyang, dry continental climate) and the boreal/hemiboreal forest biome (Helsinki). We refer to these study sites (see aerial photos in Fig. S1 in SM) by the first three letters of the city name and a sequential number. Sites in Baltimore and one of the sites in Helsinki (HEL3) were located on the outskirts of the respective cities along highways where open fields are adjacent to dense forests. In contrast with Baltimore and HEL3, the Shenyang sites were located in two urban parks where one transect was in mostly open area and the other in mostly tree-covered area (Fig. S2 in SM). However, the structural difference of the transects in Shenyang is well indicated by the average wind measurements that resulted ca. 4 times stronger wind speeds at the open transects compared to forest transects. In addition to traffic-derived pollution, these urban parks were exposed to pollution sources other than the main street, including occasional vehicles inside the parks, 2-stroke string trimmers and cooking stands, potentially adding uncertainty to the transect comparison. Photographs facing the forest transect from the road are available in SM Figure S3.

Description of the studied landscapes are provided in Table 2. Canopy closure in forests was estimated by calculating the area covered by vegetation from 1–5 skyward facing photographs per sampling point, taken 1.5 m above the ground, and averaged to give a single canopy closure estimate per site (Table 1).

2.3. Sampling

For a simultaneous comparison between open and forest transects, two particle counters were used at the same time, rotating among stations in both transects in synchrony. Sampling was carried out at four points along each transect at the height of ca. 50 cm (Fig. 1, see SM Fig. S2 for detailed site aerial photographs). The first location on the forest transect was always at the forest edge and on the open transect at an equivalent distance from the edge of the

Table 2
Description of the studied open and forest transects with dominant species and other relevant information.

	Open	Forested	Dominant species	Other
Baltimore				
BAL1	Agricultural field	Dense, deciduous-dominated natural forest patch	All BAL-sites were typified by oaks (*Quercus* spp.) and tulip trees (*Liriodendron tulipifera*)	
BAL2	Meadow	Dense, deciduous-dominated natural forest patch		
BAL3	Meadow	Dense, mixed natural forest patch		
Helsinki				
HEL1	Sports field	Dense, mixed natural forest patch	All HEL-sites had *Picea abies, Pinus sylvestris, Sorbus acuparia, Acer platanoides, Betula pendula, Populus tremula*, but HEL3 had deciduous trees only at the road verge	At HEL1 the open landscape was ca. 6 m lower than the forest landscape and beside the road was ca. 60 cm high concrete wall (at both transects)
HEL2	Park area	Dense, mixed natural forest patch		
HEL3	Agricultural field	Dense, conifer-dominated natural forest patch		
Shenyang				
SHE1	Relatively open park area	Mixed mature park forest	Pinus sp.	SHE1 had vegetation right next to the street at both transects, composed with low bushes (50 cm high) and row of street trees
SHE2	Relatively open park area	Mixed young trees (ca. 5 m in height)	Deciduous trees close to the street, conifers further down the transect	

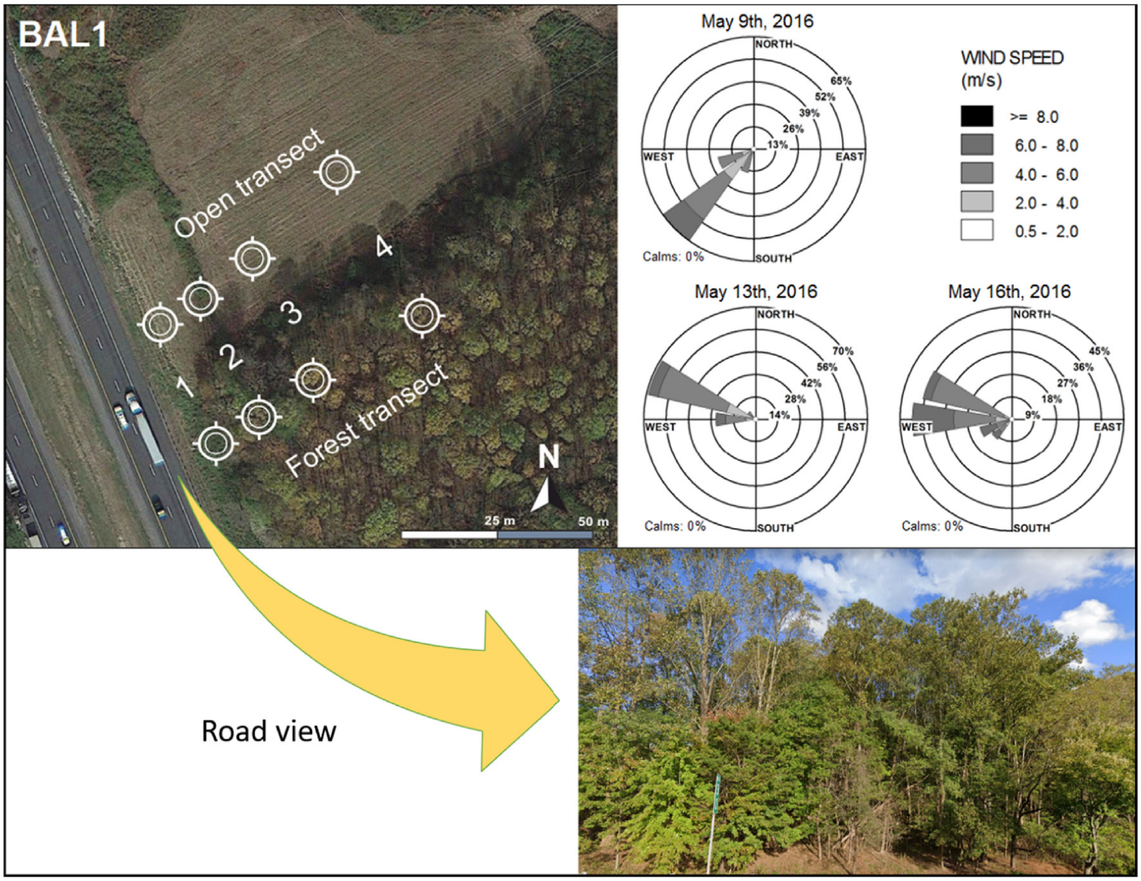

Fig. 1. The basic sampling arrangement with open and forested transects, each with 4 sampling points, and view from the road to the forest, using site BAL1 as an example. Wind roses at the site verify that during the sampling the wind was blowing from the road to the transects. © Google Earth, 2018.

road. The edge of the forest served as an anchor point to determine the other sampling locations, which were usually 10, 25 and 65 m from the forest edge (see detailed listing of the sampling distances in Table 1). The sampling distances at SHE2 differed most from those at the other sites, but the general site set-up and relative distances were similar among all sites.

The duration of sampling at each point along the transects was 10–15 min and referred to as sampling "session". The four sessions – excluding data when moving between sampling locations – including all the locations on the transect is referred to as a "cycle". Several cycles were accomplished at each site. Details on sampling dates and duration of sampling periods and cycles are presented in SM (Table S1).

Transect location for a given sampling period was selected according to predominant wind direction so that the transects were downwind of the road. Wind data from local weather stations were provided by the Finnish Meteorological Institute FMI (Helsinki) and Weather Underground (Baltimore and Shenyang). Local wind speed – averaged from all sampling sessions – at the sampling locations was measured using two portable anemometers (Trotec BA05, Trotec Group, Heinsberg, Germany). Solar radiation data, provided by FMI (Helsinki) and AERONET (Baltimore), were acquired in order to explore associations between radiation and particle levels. Solar radiation data were not available for Shenyang. The effect of wind speed and solar radiation on PNC or PNC difference between open and forest transects was studied by obtaining session averages from the PNC data and studying their potential correlation with corresponding solar radiation and wind data. Solar radiation data was also used to divide the PNC data separately for low (<550 W/m^2) and high (>550 W/m^2) radiation periods. Session data were analyzed to find out if there were differences in radiation during periods when PNC was higher in the forest than in the open vs. sessions when PNC was lower in the forest.

All the sites were sampled during the summer leaf-on period except the conifer-dominated site HEL3, which was sampled in spring just before leaf out of deciduous trees. Average temperature during sampling periods ranged between 9 and 27 °C. Average temperatures and general notes of the weather during sampling hours are provided in SM (Table S1). Traffic volumes (Table 1) were obtained from the Maryland Department of Transportation (annual average daily traffic) and Finnish Transport Agency (annual average daily traffic; annual average weekday traffic). Traffic data were not available for Shenyang. Additional on-site traffic calculations were conducted, and these indicated that traffic in Shenyang was, on average, more intense (95 vehicles/min) than in Baltimore and Helsinki, which averaged 68 and 42 vehicles/min, respectively (see Table 1 for more detailed on-site calculations).

2.4. Statistical analyses

Analysis was performed using R version 3.6.3 (R Core Team, 2020). The statistical analysis of data in studies with large amounts of high-frequency data but a limited number of replicates ("site" being a valid replicate in this case) is challenging. However, the eight sampling sites in the three cities allowed us to apply a statistical model in which site averages obtained over several sampling periods were used to run a general linear mixed effects model (GLMM) to study the effects of vegetation cover and distance on

PNC.

Data analyzed by the model were modified so that the background concentrations for each sampling period were subtracted and the calculated site means were normalized to the value obtained from the near-road sampling point in the open transect, which served as an estimate of general ambient pollution level at that particular road. Normalization was conducted by dividing all other mean values of that specific site by this value. Background was calculated as 25th percentile of data values from the sampling points located furthest from the road (average of open and forest sampling point). Normalized PNC variable was modeled (using GLMM) against the effects of distance (classified as 1–4 in the model) and transect, with site included as a random term, nested in city (to take potential spatial variation produced by site and city into account). While normalization was necessary in order to run the model, it should be noted that because this modification equalizes the differences in the background PNC levels among the sites, the model cannot control for aerosol processes regulated by ambient concentration.

Wind speed and its difference between the open and forest transects were analyzed using a paired samples *t*-test and general linear model (ANOVA) using IBM SPSS Statistics 24.

In addition to mean values, the peak events were analyzed using the Gumbel Method (Gumbel, 1941) as in Whitlow et al. (2011) to see if return frequencies of extreme events differ among transects. This is the same method used to analyze return frequency of floods and storms of a given magnitude except that the time scale is in seconds instead of years. Return frequencies provide an estimate of exposure risk at different locations.

3. Results

To provide a general overview of the influence of vegetation on near road nanoparticle concentration trends, we first present pooled data from all three cities (Fig. 2). Detailed data are presented mostly as Supplementary Data.

In the pooled data, there was a slight but statistically significant ($p = 0.039$) difference in mean PNC between open and forested transect (Fig. 2b and Table S3 in SM). As described in Materials and Methods, the statistical test was performed using the normalized and background subtracted values from each site (Fig. 2b) and not the actual means (Fig. 2a).

The PNC levels in the forest were lower only in Helsinki – and even there for only 2 of the 3 sites. Forest cover had no effect on PNC in either Baltimore or Shenyang (Fig. 2c). Though non-significant, when transect distances were pooled by cover type, Baltimore PNC levels were slightly (2.3%) reduced in the forests compared to open. In Shenyang, PNC levels were elevated in the forests by 3.7%, while in Helsinki the concentration reduction in the forests was 15%, on average (Fig. S4 in SM; see SM Fig. S5 for a figure showing city medians at different distances). The percentages are calculated for the actual concentrations (SM Fig. S4), not for the modified values shown in Fig. 2c.

The model in which potential effects of city and site were removed revealed that distance from the road was the most significant factor affecting PNC (Table S3 in SM). PNC declined ($p < 0.001$) with distance from the road and the trends were similar for both open and forest transects.

Decay curves plotted for adjusted values show most obvious reduction in forest for sites HEL1 and HEL3, but also to lesser extent for Baltimore sites (Fig. 3). PNC was higher at the forest edge compared to the equivalent distance at the open transect in half of the sites. Boxplots for sites and separate sampling periods show that sampling periods within a site differed to some extent (Figs. S6–S9 in SM). Average PNC was distinctly different among the three cities, with Helsinki (ca. 10,000/cm^3) < Baltimore (ca. 20,000/cm^3) < Shenyang (ca. 40,000/cm^3) (Fig. S4 in SM).

Real-time (1-s sampling frequency) data plots revealed occasional higher, delayed and prolonged peak events at the furthest distances in the forest compared to open transect, but these observations were inconsistent (Fig. S10 in SM). Return interval graphs showed most consistent difference in peak recurrence patterns between open and forest transects in Helsinki (see Fig. S11 in SM for HEL1 sampling periods), but not for other cities. For example when comparing two sampling periods with similar meteorological conditions at SHE2, the largest peak events were higher in the open while only on 23rd of May the bulk of the medium-sized events were higher on the forest transect (Fig. S12 in SM). In general, the data clearly show that peak events are smaller and occur less frequently with increasing distance from the road. Note that these graphs visually emphasize a small number of rare, peak events at a particular location. This is less apparent in the bulk of the data which influence averages due to overlap and the logarithmic scale (comparison of SHE2 return periods with median boxplots in SM Fig. S9 will show that highest peak events do not necessarily imply higher median).

3.1. Local wind and solar radiation

Local wind speed was clearly ($p < 0.001$) lower in the forested

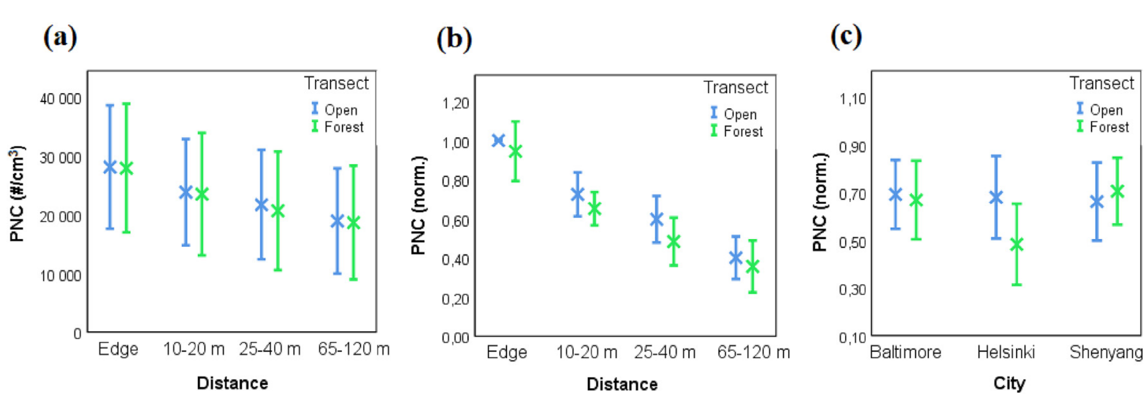

Fig. 2. PNC (mean ± SE) at open (blue) and forested (green) transects before (a) and after (b) background subtraction and normalization. Edge refers to forest edge (or on the open transect to equivalent distance from the road) from which the further distances are measured. Panel (c) shows background subtracted and normalized PNC levels (mean ± SE) levels in Baltimore, Helsinki and Shenyang pooled by cover type. Data from all three cities are pooled together for (a) and (b) (total n = 8; n = 3 for HEL and BAL, and n = 2 for SHE). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

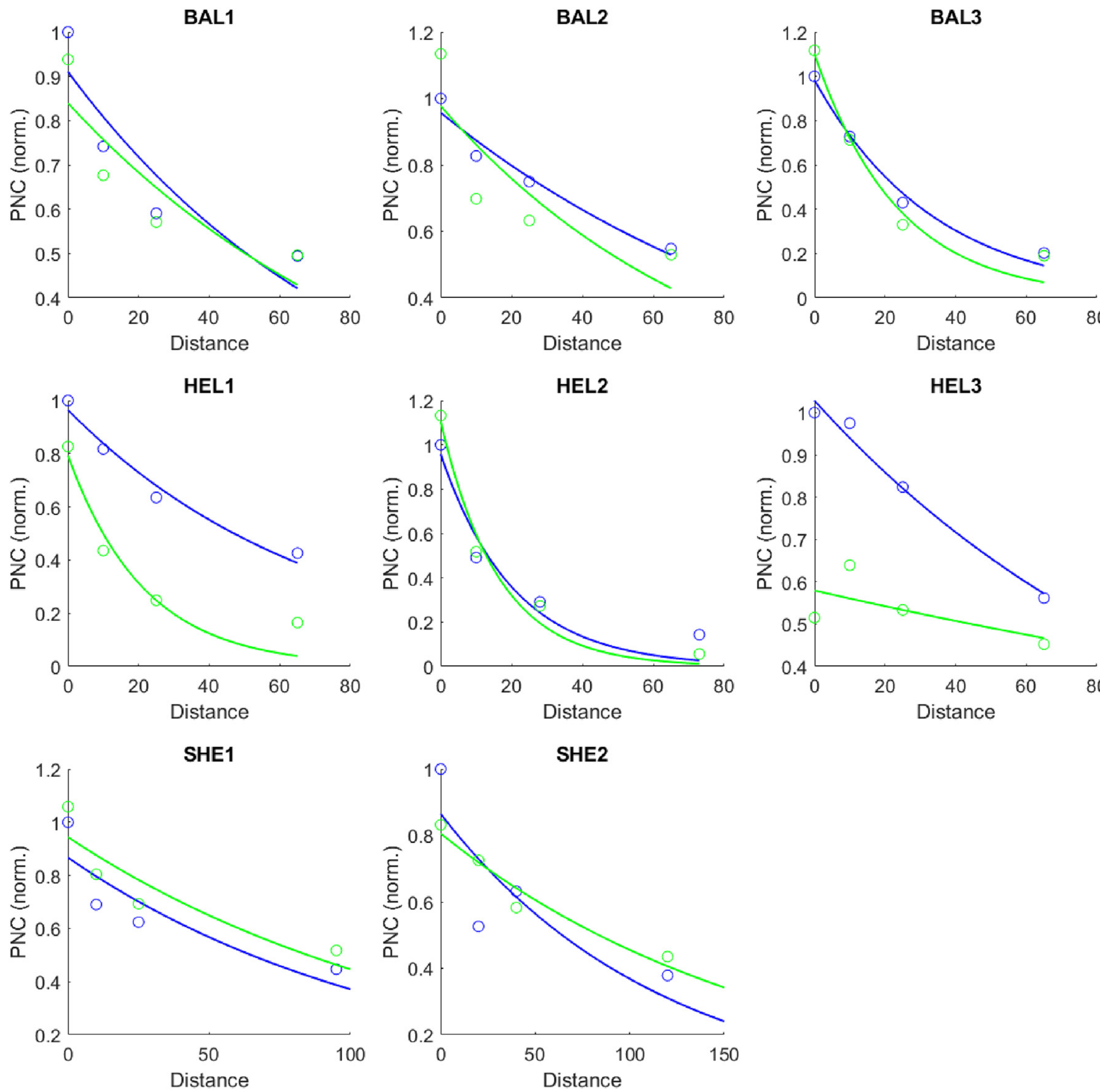

Fig. 3. Average PNC as a function of distance (meters from the forest edge facing the roadside or equivalent distance in the open) with exponential decay curves fitted for open (blue) and forested (green) transects at each site. Data is background-substracted and normalized to the conditions nearest the road on the open transect. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

transects compared to the open transects (Fig. 4), being on average ca. 4 times higher at the open transect. Distance from the road affected the wind speed difference between open and forest sampling points so that wind speed differences were greater with increasing distance into the forest ($p = 0.005$; result not shown). This was due to the wind decrease in the forest but also because the wind increased in the open with distance from the road. Wind roses based on the local weather stations data are shown in Fig. 1 (BAL1) and in Figure S2 in SM (all sites).

Neither solar radiation nor local wind speed correlated with PNC levels or PNC level differences between open and forested landscapes at the study sites (data not shown). This was true also when PNC data was treated as subgroups of high (>550 W/m^2) and low (<550 W/m^2) radiation levels.

4. Discussion

Our study focusing on nanoparticles (NP) > 10 nm in near-road environment at three geographic locations showed no clear overall impact of vegetation on particle number concentrations (PNC). A clear and statistically significant reduction was observed only in Helsinki, while in Baltimore and Shenyang the average levels were only slightly lowered and elevated, respectively.

Our three initiating hypotheses stating: i) near road PNC will be higher in the forest; ii) further from the road the particle levels will be lower in the forest and; iii) decay curves of NPs will be steeper in the forest, were only weakly supported by our findings. Half of the sites showed higher particle concentrations at the forest edge, and only in HEL1 and HEL3 PNC levels were markedly reduced further away from the road. Basically BAL2 was the only site showing good agreement with all three hypotheses, although with lesser reduction. Furthermore, neither vegetation type nor traffic density univocally influenced the role of vegetation in NP removal at near-road environments. However, while in Helsinki the reduction was clear at sites with highest canopy closure (HEL1: 77% and HEL3: 75%), in Baltimore, where the reduction was not evident, canopy closure was even higher (85%). The reason for this deviation was not explained by any of the measured variables. Yet in Shenyang,

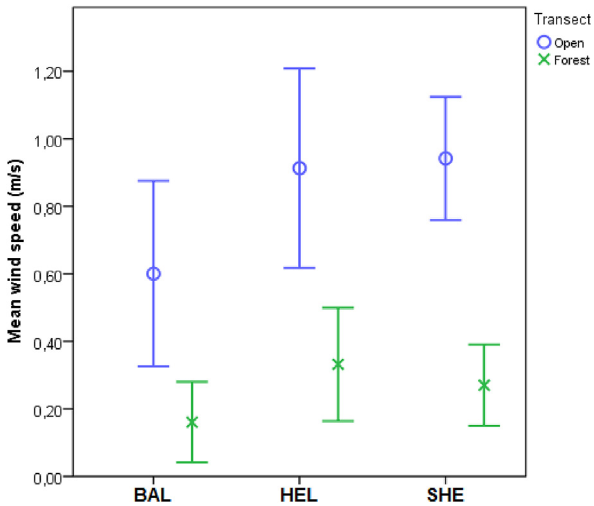

Fig. 4. Local wind speed (mean ± SE) at the open (spherical symbols) and forested (cross symbols) landscapes in Baltimore, Helsinki and Shenyang. Distances are pooled together.

where the canopy closure was smallest, the levels were higher in the forest. These observations are to some extent in line with the studies showing vegetation density to improve its performance in reduction (e.g. Baldauf, 2017; Deshmukh et al., 2019). Differences between Baltimore and Helsinki suggest that other conditions may also play a role, such as climatic factors or differing ambient particle concentrations. These findings support recent observations (Abhijith et al., 2017; Hagler et al., 2012; Janhäll, 2015; Vos et al., 2013; Xing and Brimblecombe, 2019; Yli-Pelkonen et al., 2017a) indicating that tree arrangement, dimensions, species composition and other variables affect whether the presence of forest cover improves local air quality.

Tailpipe pollution – which is usually the most important source of NP pollution in urban areas – is emitted close to ground level, where atmospheric flow, and thus pollution dispersion, is affected by strong turbulence and a variety of physical structures, all of which increase randomness in pollution gradients (Enroth et al., 2016). Thus it is obvious that tree/vegetation effect on NP decay gradients will be different in street canyons, behind vegetation barriers, and in the locations studied here. However, numerous studies in urban street canyons (see Gromke et al., 2016 and references therein) have documented urban trees to reduce air exchange and thus result in elevated pollution concentrations at the street level. Although there are many studies of urban vegetation and air quality, most address street canyons or roadsides with deliberately constructed vegetation and sound barriers (e.g. Barwise and Kumar, 2020; Ghasemian et al., 2017). Studies like ours where potential trapping of NPs by trees is explored further from roads within a continuous tree cover are rare. For instance, Hagler et al. (2012) measured traffic-derived NPs behind vegetation barriers and reported that the effect of vegetation was inconsistent and ambiguous. However, Brantley et al. (2014) found that a vegetation barrier reduced black carbon concentrations but had no effect on particles > 500 nm. In contrast, Al-Dabbous (2014) found 37% percent lower PNC behind compared with in front of a vegetation barrier suggesting that the barrier deflected or trapped small-sized particles. Similarly, Baldauf (2008) detected lower PNC behind a solid noise barrier with mature trees in its wake compared to a solid barrier without trees or open terrain. In each of the former studies particulate concentrations were reduced only in cases where wind was blowing across the road towards the barrier. In our study, the predominant wind direction during measurements was from the road to the transects, but not always perpendicularly, as shown by the aerial photos with wind roses. This will complicate the flow fields and result in some of the unsystematic behavior between the sites and sampling sessions. It should be emphasized that our study design is not directly comparable with aforementioned barrier studies, where air pollutants were measured behind the barrier as opposed to within the tree stand as in the current study. Surprisingly, only studies by Hagler et al. (2012) and Deshmukh et al. (2019) included more than one site, reflecting the strong site-specific approach in virtually all earlier PNC studies.

4.1. Wind speed

Our result suggests that the effect of wind speed in determining PNC levels in open and forested transects is less important than expected, or that there are other confounding factors like varying wind direction that are more important than expected. Here wind patterns were basically the same at every site, wind speed being ca. 4 times higher in the open landscape, yet forest performance differed among sites. According to a review by Janhäll (2015), in order to derive the greatest benefit from particle deposition onto vegetation surfaces, vegetation should be placed near the pollutant source and the vegetation structure should be porous enough to allow the air mass to pass through, otherwise pollution will not reach the vegetation surfaces inside the vegetation patch. However, reduced ventilation may elevate concentrations inside vegetation stands even when deposition is significant (Janhäll, 2015). Wind speed measurements at the local scale are thus needed to assess how porous forest patches are, i.e. how efficiently the polluted air enters the forest area and how much of it is dispersed through the open area.

4.2. Factors affecting the ability of trees to capture particles

The most important processes by which trees are considered to affect particle levels are particle deposition on tree surfaces and trees' effect on dispersion, yet a number of other factors may affect particle concentration. NP concentrations and the shape of the pollutant gradient next to roads depends, in a complex manner, on local meteorological conditions (wind speed, wind direction, temperature, boundary layer height) and aerosol transformation processes (nucleation, coagulation, condensation, evaporation and deposition) (Enroth et al., 2016; Kumar et al., 2011). These factors occur simultaneously, making it challenging to deduce the most influential variables that affect PNC at a given time and place. This may in part explain the lack of clear and all-encompassing effect of trees on particle levels in the three geographic regions in our study. Consequently, knowledge of the formation and growth of NPs are essential for understanding aerosol dynamics. After being emitted as exhaust or nucleated from gases immediately after being emitted, NPs are exposed to transformation processes that may be different under the canopy and in open areas – mainly due to differences in exposure to solar radiation, humidity, biogenic VOC levels (from vegetation) and air movement.

4.3. Peak events

Near-road environments are typified by emissions from the passing vehicles that stand out as multitude of peak particle events superimposed on a more stable background particle level that may change during the day. Viippola et al. (2018) observed differences between open and forest transects among highest peak events for $PM_{2.5}$ and coarse PM, where peak events in the forest transect were higher near the forest edge and lower further away from the road compared to the open transect. In the current study no such pattern was observed, suggesting that the behavior of NPs differs from that of $PM_{2.5}$. This may partly be due to the fact that NPs behave more like gas molecules and deposition rates are limited mainly by diffusion, not impaction and gravitational settling.

In the current study, the peak events were occasionally prolonged in the forest compared to open transects. This was expected, but the prolonged peak events occurred mainly further away from the road rather than next to it (Fig. S10, SM). Further, when comparing peak events in the open and forest transects, peaks often occurred earlier in the open transects. This is likely due to the aerodynamic resistance of the forest, which was occasionally detected at the furthest sampling point in the forest (Fig. S10, SM).

5. Conclusions

In this field study, no consistent forest effect for nanoparticles (>10 nm) was found. This is in contrast with some earlier studies – most of which rely on only one study site – where nanoparticle reductions due to trees were observed during crossroad winds.

Discussion around vegetation in improving air quality often falls short of elucidating the full implications of any given mitigation scenario. While e.g. 50% pollution reduction in a small zone behind a 10 m vegetation barrier may reduce pollutant exposure at that particular location, it is important to acknowledge that only if the pollutants are actually *deposited* instead of merely deflected, there is even a theoretical possibility of improving the air quality in the larger urban atmosphere. Furthermore, in order to maximize pollutant deposition on vegetation, the concentrations around vegetation must be very high (Janhäll, 2015). This would mean that areas with vegetation aimed for pollution mitigation should be designed to restrict dispersion as much as possible, which would turn the areas into no-go sacrifice zones. Reasonable applications of vegetation as a pollution sink in urban areas become here very challenging, especially as the little space left for urban green in the growing urban areas have been shown to be particularly valuable in providing cultural ecosystem services (Gómez-Baggethun and Barton, 2013), which most often require people to be able to access the green areas.

Modeling of local scale pollutant dispersion and transformation is evolving and likely to provide sophisticated means for a thorough understanding of the spatial and temporal occurrences of pollutants (e.g. Kurppa et al., 2018). However, we propose that gathering empirical pollution data using well-designed experiments under various environmental conditions is necessary in order to design plantings to improve local air quality and accurately tune simulation models, and these efforts ultimately will produce better knowledge on where and how investing on green infrastructure as a means to improve air quality is justified and feasible.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2020.115294.

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